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Assessment of soil erosion risk in a typical Mediterranean environment using a high resolution RUSLE approach (Portofino promontory, NW-Italy)

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ABSTRACT

Particularly the Liguria region in Northern Italy is highly affected by soil erosion processes. This study was conducted in the Portofino promontory in eastern Liguria, to predict potential annual soil loss using the Revised Universal Soil Loss Equation (RUSLE). Moreover, we evaluate the relative accuracy of the predictions at detailed scale, using high resolution spatial information for model calibration. The RUSLE factors were calculated for the study area based on terrain survey data and rain gauge measurements. The results were plotted on a 1:10,000 scale soil erosion map and subsequently compared with the European soil loss estimation method (RUSLE2015) developed by the European Joint Research Centre. This study shows that the RUSLE2015 model can be applied in a typical Mediterranean environment such as the Portofino promontory. However, the accuracy of the single factors we calculated using high resolution data sets might improve the results substantially and thus, also model efficiency.

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Soil loss; water erosion; GIS; modeling; Apennines

1. Introduction

Water erosion represents one of the most important and widespread causes of soil degradation in Europe (Alcamo, Florke, & Maerker, 2007). The Mediterranean region is particularly prone to erosion. This is due to long dry periods followed by heavy erosive rainfall events, and steep slopes with erodible soils, resulting in considerable amounts of eroded material (Kepner, Rubio, Mouat, & Pedrazzini, 2006).

About 77% of Italy is at risk of accelerated water erosion (Gazzolo & Bassi, 1961) with an average soil loss of $7.4 \text{ ton ha}^{-1} \text{ yr}^{-1}$ (Stolte et al., 2016). In particular, the Ligurian region (North-West of Italy), is characterized by rural areas, that cover almost 94% of the total surface (European Commission, The Rural Development Programme 2014–2020). In this region steep slopes and the abandonment of conservation measures such as terraces and waterways (Chisci, 1986), increase significantly the risk of soil erosion (Cevasco, Brandolini, Scopesi, & Rellini, 2013) and flooding (Scopesi, Maerker, Bachofer, Rellini, & Firpo, 2017, 2012).

A number of projects addressed to assess the risk of soil erosion at national, European and International level. Several multidisciplinary models have been developed to study soil erosion processes and their dynamics. In particular, the Joint Research Centre (JRC) conducted research over the last 15 years developing a series of pan-EU soil erosion assessments based

on modeling studies such as Universal Soil Loss Equation (USLE) (Van der Kniff, Jones, & Montanarella, 1999), MESALES (Le Bissonnais, Montier, Jamagne, Daroussin, & King, 2002), PESERA (Kirkby, Irvine, Jones, Govers, & Team, 2008). A new assessment of soil loss by water erosion in Europe has recently been published (Panagos et al., 2015a), aiming at policy makers. The assessment is based on the Revised Universal Soil Loss Equation (RUSLE) and several methodological and conceptual improvements were included with respect to previous JRC modeling attempts (Panagos, Meusburger, Ballabio, Borrelli, & Alewell, 2014, 2015b, 2015c, 2015d).

The well-known Universal Soil Loss Equation (USLE) (Wischmeier & Smith, 1978) and its revised versions (RUSLE) (Renard, Foster, Weeies, McCool, & Yoder, 1997), was used because it is one of the least data demanding erosion models that has been developed and it has been applied widely at different scales. The USLE is a simple empirical model, based on regression analyses of soil loss rates on erosion plots developed in the USA and subsequently applied also in a variety of other countries and regions. The model is designed to estimate long-term annual erosion rates particularly for agricultural land use. Although the equation has many shortcomings and limitations, it is widely used because of its relative simplicity and robustness (Desmet

& Govers, 1996). Some authors criticized the model structure (e.g. Evans & Boardman, 2016). They argue that for some regions this USLE based assessment of water erosion does not reflect reality as seen in the field, and if applied to slope, catchment up to regional scales and in different environments it is limited concerning input data and accuracy of quantitative outputs. Panagos et al. (2016) reply that the RUSLE is not able to reproduce 'reality', like probably any other approach, thus it cannot directly be compared to field assessments of soil loss. The intention of RUSLE is to estimate average long-term soil loss rates by sheet and rill erosion. Doing this it helps implementing soil conservation policies carried out at EU level, since local methodologies may suffer from a large heterogeneity, ambiguity, low consistency and, in many regions, a total lack of information. Moreover, the USLE modeling approach should be rather employed for comparative purposes and not considered in absolute terms (Šúri, Cebecauer, Hofierka, & Fulajtár jun, 2002). This paper illustrates the application of the RUSLE model in the Portofino regional park, a typical Mediterranean environment in Liguria Region (north-western Italy), following several methodological improvements of the new JRC modeling assessments such as the modeling of rainfall erosivity (R factor) based on rainfall intensity, frequency, amount and duration (Borrelli, Diodato, & Panagos, 2016; Panagos, Borrelli, & Meusburger, 2015b). The results of this detailed study Main Map were compared with those from RUSLE2015 for Europe (Panagos et al., 2015a), in order to test models' suitability. We applied the

RUSLE that requires modest data input to evaluate the relative accuracy of the predictions at detailed scale. For model calibration we used high resolution spatial information. In fact, the difficulties associated with calibrating and validating spatially distributed soil erosion models are, to a large extent, due to the large spatial and temporal variability of soil erosion phenomena and the uncertainty associated with the input parameter values used by models to predict these processes (Jetten, Govers, & Hessel, 2003).

2. Materials and methods

2.1. Study area

The Portofino Promontory is located on the East Ligurian Riviera in north-western Italy, and opens up towards the sea for about 3 km, interrupting the even coast line (Figure 1). The spatial extent of the promontory is about 18 km². This coastal sector generally represents a territory of great environmental and touristic value, which has been protected since 1935 in form of the Portofino Park. Seawards the area became a marine reserve in 2001. The elevation of the area range between 0 and 600 m a.s.l.. The lithology is dominated by sedimentary rocks which are known, in the regional geological literature, as: 'M.te Antola flysh' of the superior Cretaceous period and 'Conglomerates of Portofino' dating back to Oligocene age. The vegetation differs greatly from other parts of the promontory due to its particular topography and geology. The highest parts and ridges are

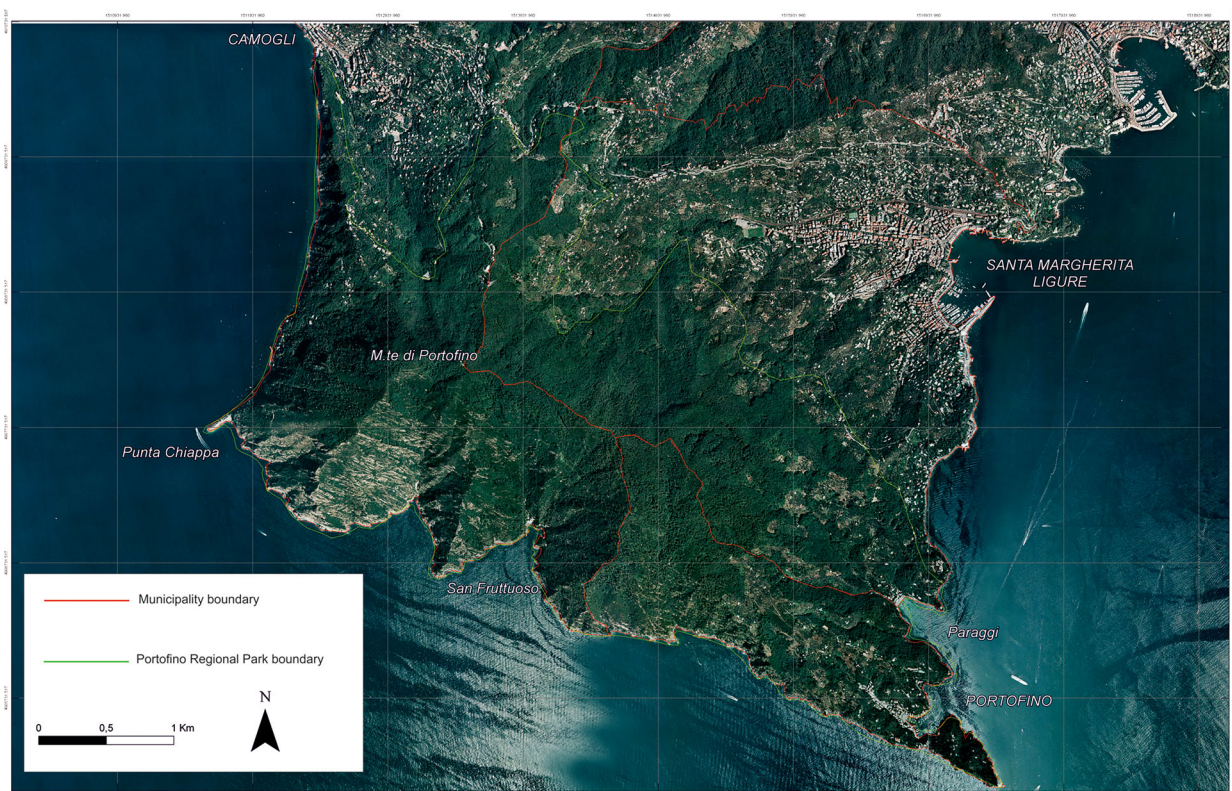


Figure 1. Study area.

characterized by woods and scrubs while Mediterranean maquis prevails on the steep slopes exposed to the south. Cultivated areas (olive groves and vineyards), on terraced slopes supported by stone walls, extend along the lowest and intermediate parts of the northern, eastern and western parts of the promontory. The territory falls into the Mediterranean humid macroclimate, with hot summers and temperate winters (Girani & Olivari, 1986). Rainfall has its maximum in autumn and its minimum in summer, with mean annual rainfall ranging between 900 and 1300 mm, mainly depending on the topography. Six Reference Soil Groups (RSGs; FAO, 2014) have been identified in the Portofino regional park: Cambisol, Regosol, Leptosol, Luvisol, Acrisol, and Umbrisol (Rellini, Olivari, Scopesi, & Firpo, 2017). Cambisols and Regosol are the dominant soils on most of the slopes, while Acrisol are prevalent in the summit areas. They are relict palaeosols restricted to a combination of relief (paleosurfaces) and deep weathering (Rellini et al., 2017).

2.2. Model and parameter estimation

The RUSLE model takes the following form:

$$A = RKLSCP$$

where A is the estimate of average annual soil loss (ton $\text{ha}^{-1} \text{year}^{-1}$) caused by sheet and rill erosion; R is the

rainfall erosivity factor ($\text{MJ mm ha}^{-1} \text{h}^{-1} \text{year}^{-1}$); K is the soil erodibility factor ($\text{t ha h ha}^{-1} \text{MJ}^{-1} \text{mm}^{-1}$) which is a measure of the susceptibility of soil to be eroded under standard conditions; LS is the topographic factor, derived from a combination of slope steepness and specific catchment area (non dimensional); C is the cover and management factor (non dimensional); P is the support practice factor (non dimensional).

The advantage of a selection of RUSLE is that the parameters of this model can be easily managed and visualized within a GIS. The parameters of the RUSLE model were estimated based on rainfall data, DEM, soil type, and land cover. The overall methodology used in the present study is schematically represented in Figure 2 (flow chart of methodology).

2.2.1. Rainfall erosivity factor (R)

The rainfall erosivity factor (R) reflects the effect of rainfall intensity on soil erosion, and requires detailed, continuous precipitation data for its calculation (Wischmeier & Smith, 1978). In the present study, high-temporal-resolution pluviographic records were collected from 4 rain-gauges of the Regional Hydrographic Services and Regional Agencies for Environmental Protection (ARPAL). The selected stations are well distributed throughout the promontory. The time-series were composed of continuous 10-year

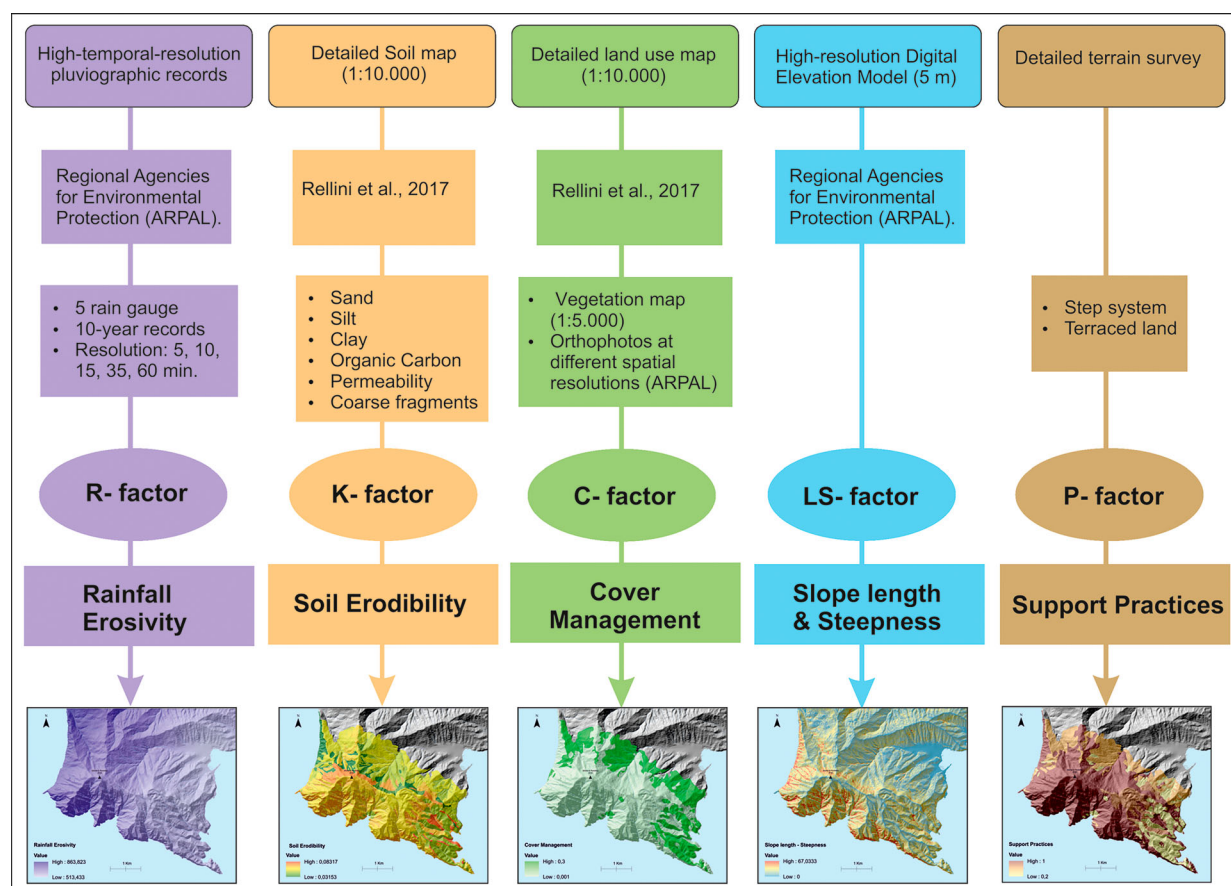


Figure 2. Input dataset used for estimation of soil loss factors for study area in RUSLE.

records for most of the stations (2002–2011). This is the minimum time period to sufficiently represent the inter-annual variability of rainfall erosivity (Borrelli et al., 2016). The stations cover the same number of recorded years, minimizing measurement errors related to the inter-annual variability of rainfall erosivity (Borrelli et al., 2016; Capolongo, Diodato, Man-naerts, Piccarreta, & Strobl, 2008). Monthly rainfall-runoff erosivity values (R -factor) were computed according to Wischmeier and Smith (1978), the rainfall erosivity index (EI_{30}) of each erosive storm between January 2002 and December 2011 is given by the product of the storm kinetic energy and the maximum 30-min intensity (I_{30}):

$$EI_{30} = \left(\sum_{r=1}^0 e_r v_r \right) I_{30},$$

where e_r is the unit rainfall energy ($\text{MJ ha}^{-1} \text{mm}^{-1}$) and v_r is the rainfall volume during a given period (r). The unit rainfall energy is calculated for each time interval according to Brown and Foster (1987):

$$e_r = 0.29[1 - 0.72e^{(-0.05I_r)}],$$

where I_r is the rainfall intensity during the time interval expressed in mm h^{-1} . Finally, the annual average rainfall erosivity is expressed as

$$R = \frac{\sum_{i=1}^j (EI_{30})_i}{N},$$

where the $(EI_{30})_i$ is the EI_{30} for rainstorm i and j is the number of rainstorms in an N -year period. Storm-by-storm summaries of precipitation, duration, intensity, kinetic energy and the rainfall erosivity index (EI_{30}) were computed using the Rainfall Intensity Summarization Tool (RIST v3.88) (USDA, 2014). The value of 1.27 mm of precipitation in six hours was selected to represent threshold rainstorms. Afterwards, rainstorms with less than 12.7 mm of precipitation were omitted from the EI_{30} calculation (Foster et al., 1981; Renard et al., 1997).

Spatial distribution of average annual R -factor in the study area is estimated using 'Kriging' as the interpolation method.

2.2.2. Soil erodibility factor (K)

The Soil Erodibility factor (K) represents the susceptibility of soil or surface material to erosion, transportability of the sediments, and the amount and rate of runoff given for a particular rainfall input as measured under standard conditions. Soil erodibility was calculated according the approximation of the soil erodibility nomograph equation (Renard et al., 1997; Wischmeier & Smith, 1978) in Equation (1):

$$K = [2.1 \cdot 10 - 4(12 - OM) \cdot 1.14M + 3.25(s - 2) + 2.5 \cdot (p - 3)] / 100 \cdot 0.137 \quad (1)$$

where OM is the percentage of organic matter in the surface horizon (equal to 4 in cases where the OM is greater than 4%); M is given by the textural equation:

$$M = (\% \text{ sand} + \% \text{ silt}) \cdot (100 - \% \text{ clay});$$

and s and p are the soil structure class and soil permeability class, respectively.

Data concerning the areal distribution of the K factor was derived from the published soil map of Portofino promontory (Rellini et al., 2017). The distribution of the K -factor values was determined by associating a representative value (benchmark profile) with each cartographic unit. For soil associations, the K -factor value was calculated based on the weighted averages of the profiles.

2.2.3. Topographic factor (LS)

LS factor was directly calculated from high-resolution Digital Elevation Model (5 m resolution) of the study area and from the following equation proposed by Moore and Wilson (1992), using a unit contributing area (UCA) to calculate LS for three-dimensional terrain and implemented in the SAGA GIS environment:

$$Ls = \left(\frac{As}{22.13} \right)^m \left(\frac{\sin \beta}{0.0896} \right)^n$$

where As is the unit contributing area (m), β is the slope in radians, and m (0.4–0.56) and n (1.2–1.3) are exponents. The DEM was based on an interpolation of contour lines from a 1:5000 topographic map (Regione Liguria, 2007) using a thin plate spline algorithm proposed by Hutchinson (1996). The DEM was preprocessed with low-pass filtering to extract artefacts and errors, such as local noise and terraces (Vorpahl, Elsenbeer, Märker, & Schröder, 2012).

2.2.4. Crop management factor (C)

The C factor was estimated on the basis of the soil map legend (Rellini et al., 2017). The values for each land use type were assigned using literature data (among others: Märker et al., 2008; Wischmeier & Smith, 1978) and are illustrated in the following table (Table 1):

Table 1. Land use percentage distribution and C factor values derived from literature.

Land use/cover	%	C _factor
Olive groves	32	0.3
Garigue	14	0.01
Maquis	37	0.005
Chestnut wood	3	0.003
Mixed forest	6	0.003
Mixed oak forest	3	0.003
Transitional woodland-shrub	1	0.002
Ilex wood	4	0.001

Table 2. *P*-factor values according to literature and Munro et al. (2008).

Support practices	<i>P</i> _factor
No practices	1
Step system 'ciglionamento'	0.8
Abandoned terraced land	0.6
Terraced land	0.2

2.2.5. Conservation support practice factor (*P*)

The conservation practice factor (*P*) represents the ratio of soil loss by a support practice to that of straight-row farming up and down the slope and is used to account for the positive impacts of those support practices. The *P* factor accounts for control practices that reduce the erosion potential of the runoff by their influence on drainage patterns, runoff concentration, runoff velocity, and hydraulic forces exerted by runoff on soil. The value of *P* factor ranges from 0 to 1, the value approaching to 0 indicates good conservation practice and the value approaching to 1 indicates poor or absent conservation practice. For the study area, the *P*-factor for terracing and step type landscapes are the only practices used in the study area other practices are absent. We used the values for the terraced land available from literature but considering the modification proposed on the quality of stone walls (Munro et al., 2008). *P*-factor values, range from about 0.2 for terraces with good quality of stone wall, to 1.0 where there are no erosion control practices, usually for natural slope (Table 2):

3. Conclusion

This study applied a GIS based analysis using the RUSLE model to estimate annual soil loss on pixel level (25 m²) and to assess the spatial distribution of soil erosion in the study area. Soil loss values range between 0 and 225 t ha⁻¹ yr⁻¹ with a mean value of 9 t ha⁻¹ yr⁻¹ Main Map. According to Pimentel et al. (1976), a threshold value for a sustainable soil loss as is amounting to 11–12 t/ha/yr. In the study area only about 25% of the cells show a soil loss value greater than the limit of 'tolerable' soil loss: 10% of the area falls into class 5 (10–20 t ha⁻¹ yr⁻¹), 8% into class 6 (20–30 t ha⁻¹ yr⁻¹) and 7% into class 7 (>40 t ha⁻¹ yr⁻¹) Main Map.

The results of this study provide useful information for decision makers and planners to take sustainable land use management and soil conservation measures in the Portofino Natural Park. The highest soil loss values are associated with the cultivated areas on the terraced slopes Main Map. Hence, substantial conservation practices should be taken into account in these areas. Land cover is the main driver for soil erosion. In fact, high and extreme erosion occur mainly in areas that show significant changes in land use, for instance: steep slope cultivation and excessive deforestation.

This study also contributes to validate the existing national data set in comparison to the soil erosion map of Europe. The results of the application of the RUSLE model are quite consistent with those obtained with RUSLE2015 for the study area, which show a mean value of 7 t ha⁻¹ yr⁻¹ (European Soil Data Centre; Panagos et al., 2015b). However, there are also distinct differences due to the higher spatial resolution of the input information. So, locally higher as well as lower values are simulated with the detailed approach in respect to the RUSLE2015 approach (Panagos et al., 2015b). Especially, *R*-factor and *C*-factor (land use) is triggering mainly the spatial distribution. Moreover, detailed information about terraced areas lead to significant lower soil erosion since runoff velocity is decreased. Consequently, mapping of these areas, frequently characterized by abandonment, with modern methodology (i.e. LiDAR survey), may be crucial to provide the best input for soil erosion analysis.

This study shows that the accuracy of the single factors due to a higher resolution RUSLE dataset improves the results and the model efficiency. This is particularly interesting and valuable for landscape planners, that have to deal with climate change effects as just recently experienced during an exceptional rainstorm event at the beginning of November 2018 (31.10.–01.11.2018). Such a detailed approach allows also for a specific assessment of different support practices and/or changes in landuse. In the prospective of more intensive and frequent exceptional events this knowledge might be very useful to prepare the area and to identify the hotspots for erosion and for Soil Organic Carbon (SOC) preservation that might need intervention measures. In fact, soil erosion can affect SOC content by direct removal of soil and reduction of the surface soil depth and that is of great importance because of global environmental concerns (Lal, 2003). However, in this study only soil erosion by surface runoff is taken into account, concentrating on rill- and inter-rill erosion. Thus, the produced maps should not be used to predict the occurrence of mass movements such as landslides and mudflows.

Software

For implementation of the research method, the following software programs were used:

- (1) ArcGis 9.3 (ESRI, 2006) is used to create the attribute database and compilation of the RUSLE map showing the potential of soil erosion rate.
- (2) Saga Gis (Saga Development Team, 2011) is used to clip DEM and to produce spatial distribution of *R* factor (using 'Universal Kriging' method of interpolation) and *LS* factor (using the algorithm proposed by open source GIS).

Disclosure statement

No potential conflict of interest was reported by the authors.

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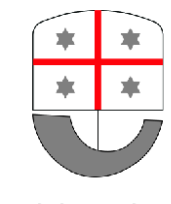
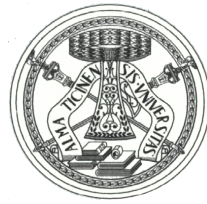
Assessment of soil erosion risk in a typical Mediterranean environment using a high resolution RUSLE approach (Portofino Promontory, Italy).

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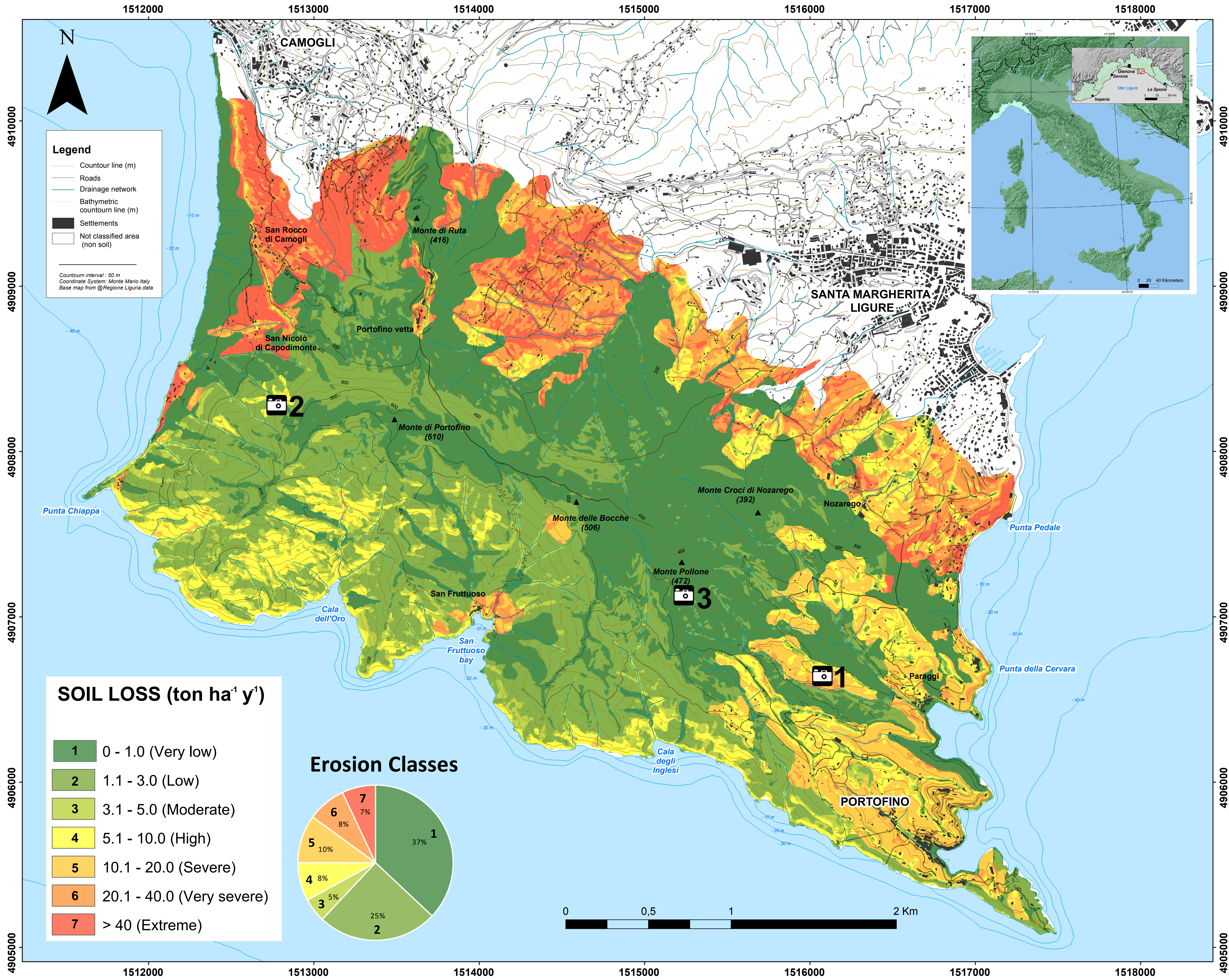
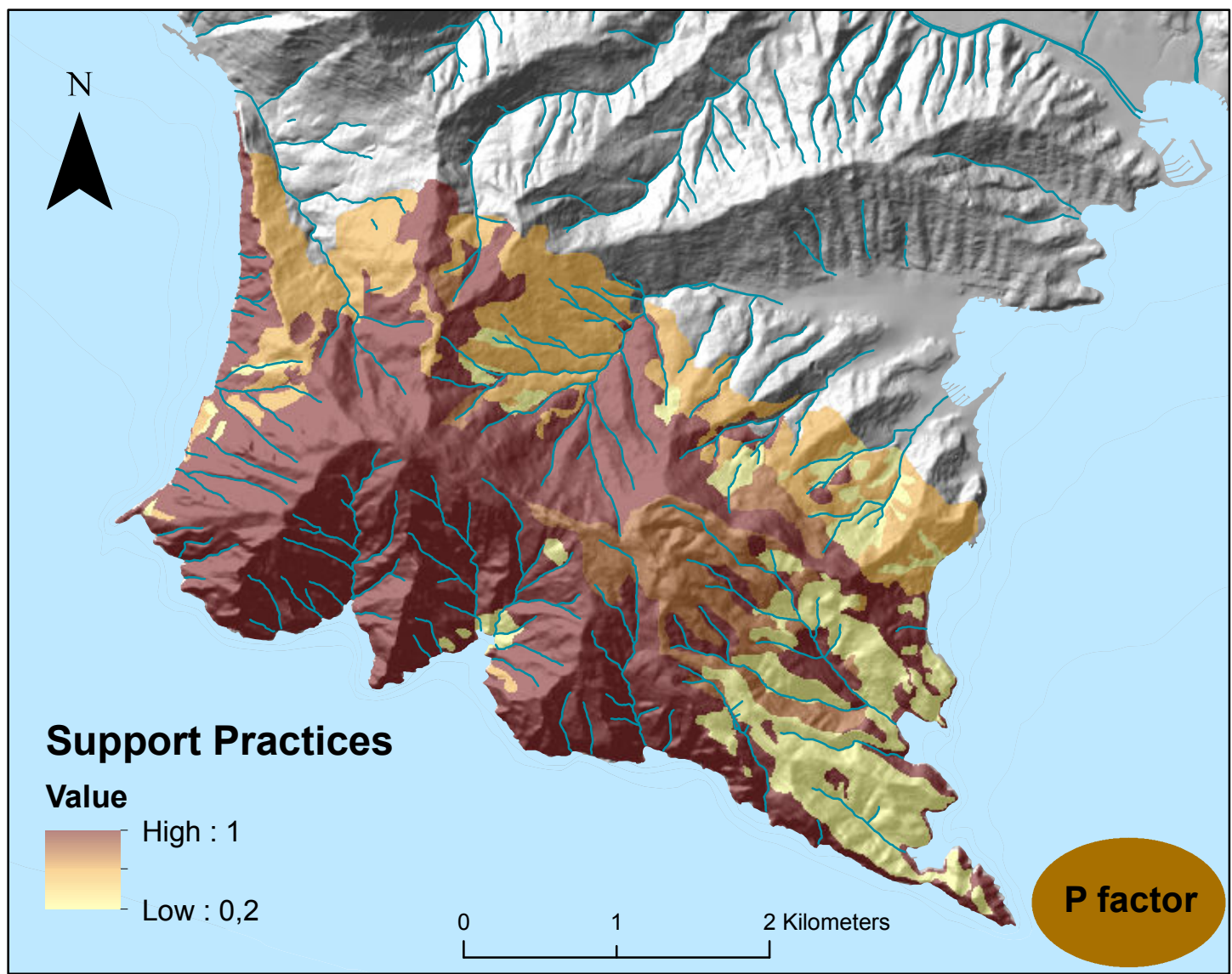
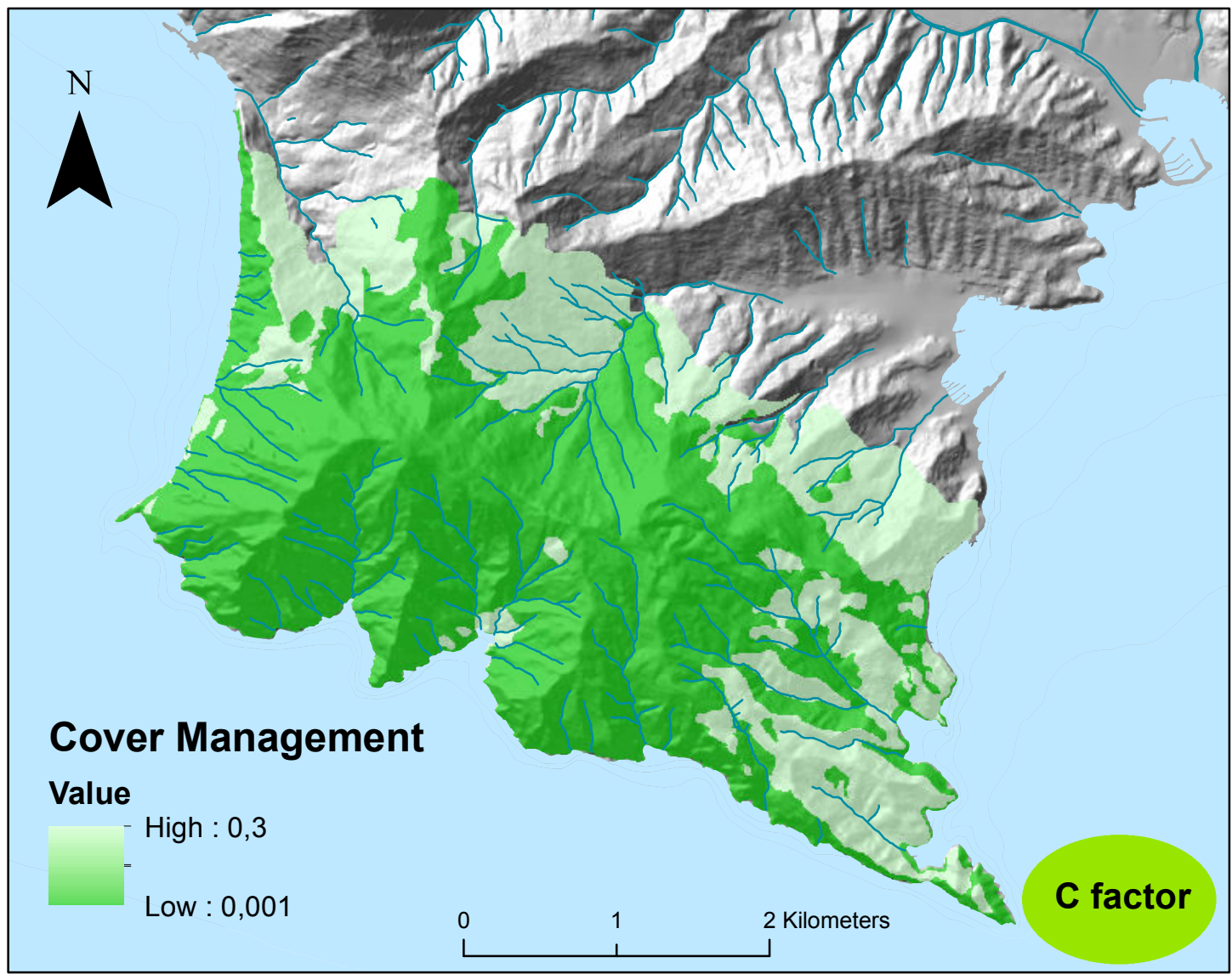
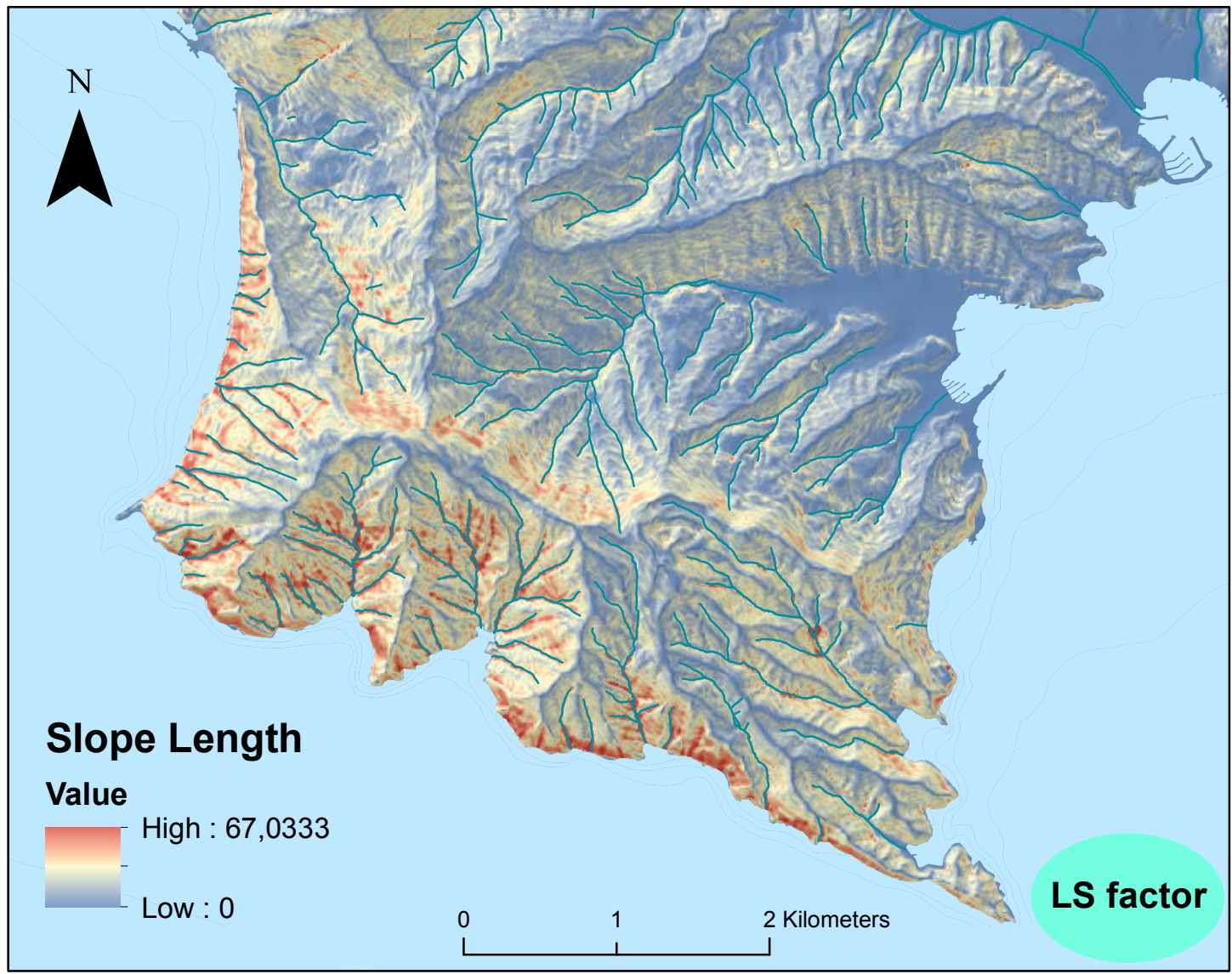
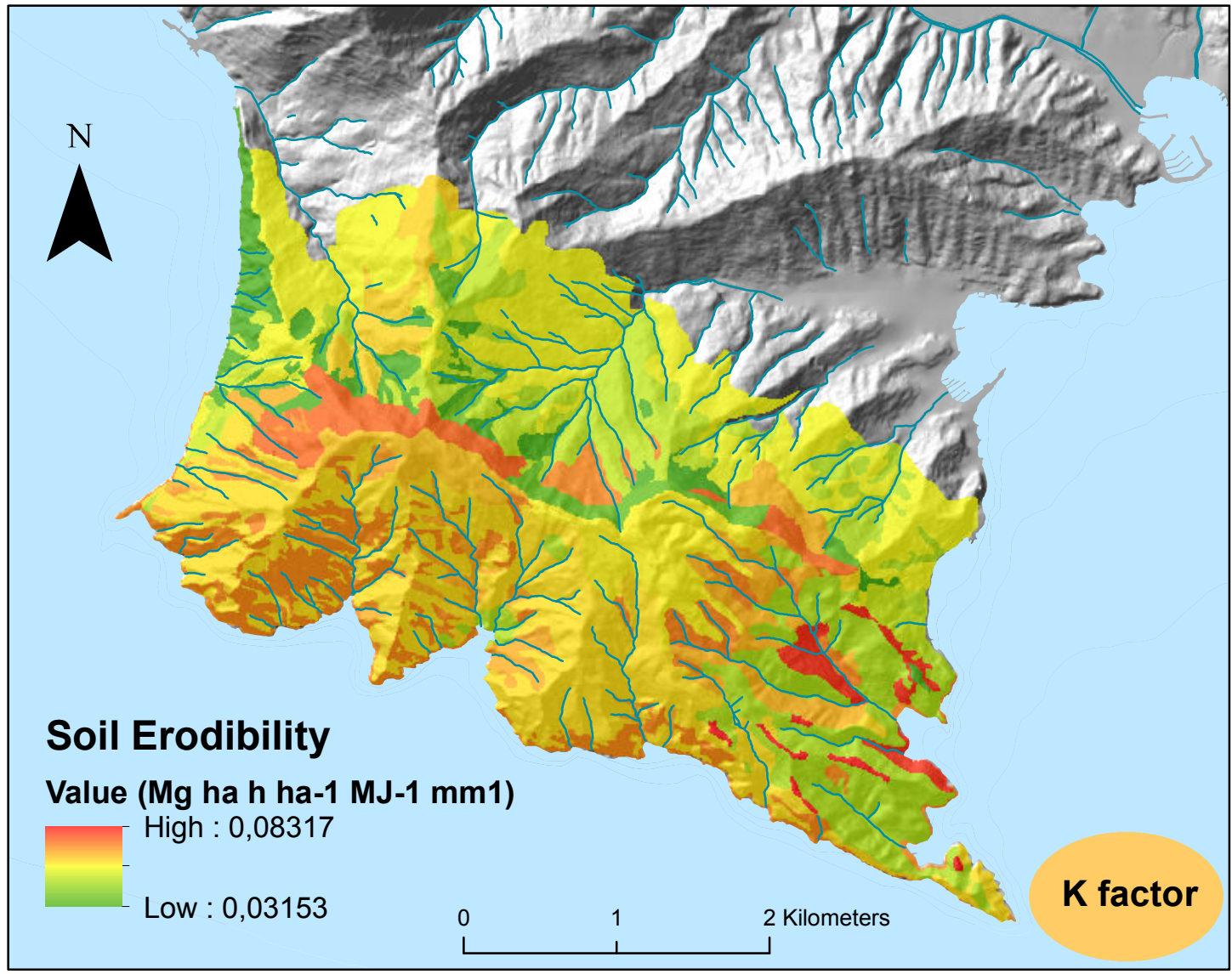
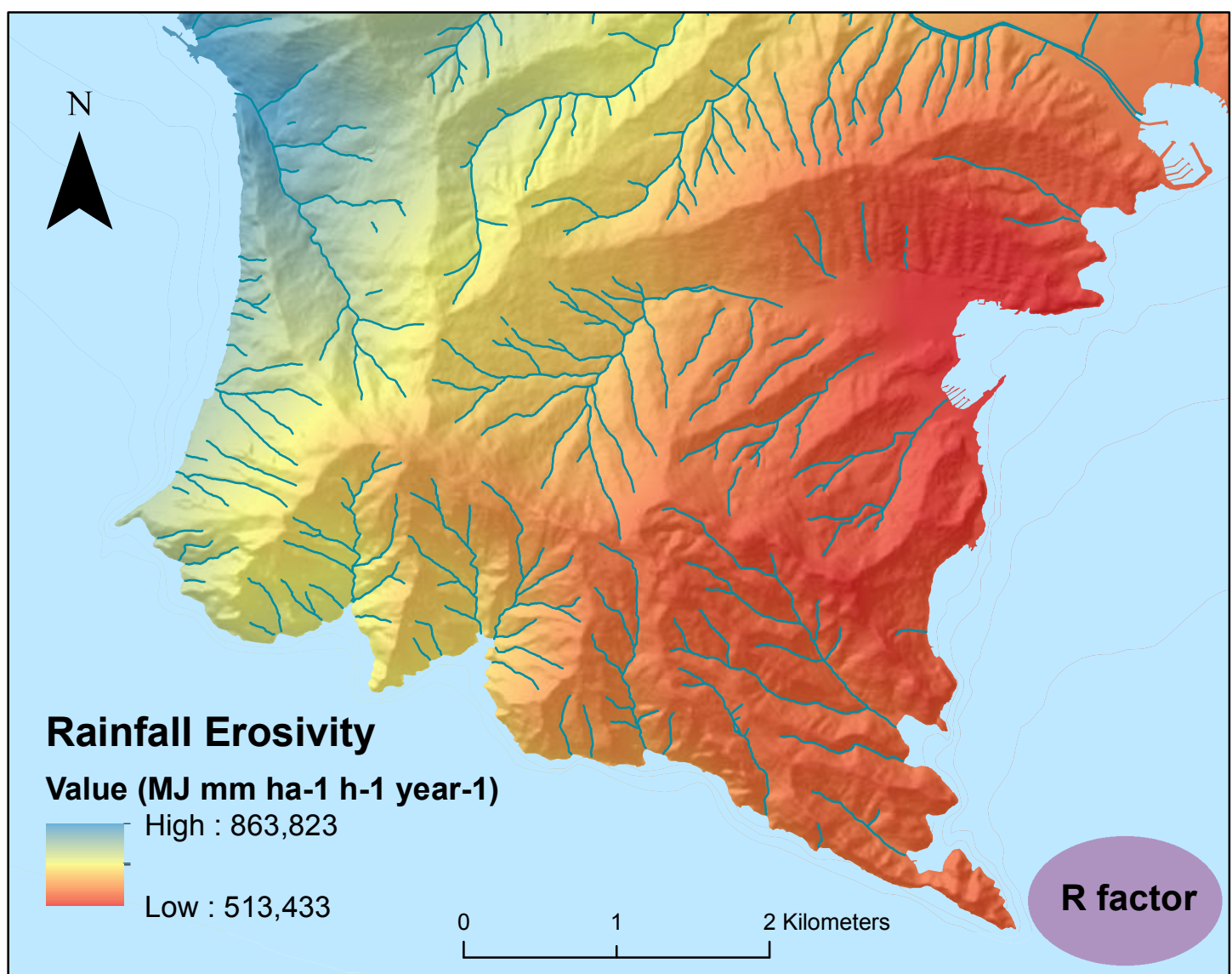
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Potential annual soil loss calculation
using the revised universal soil loss equation (RUSLE, Renard et al., 1997)

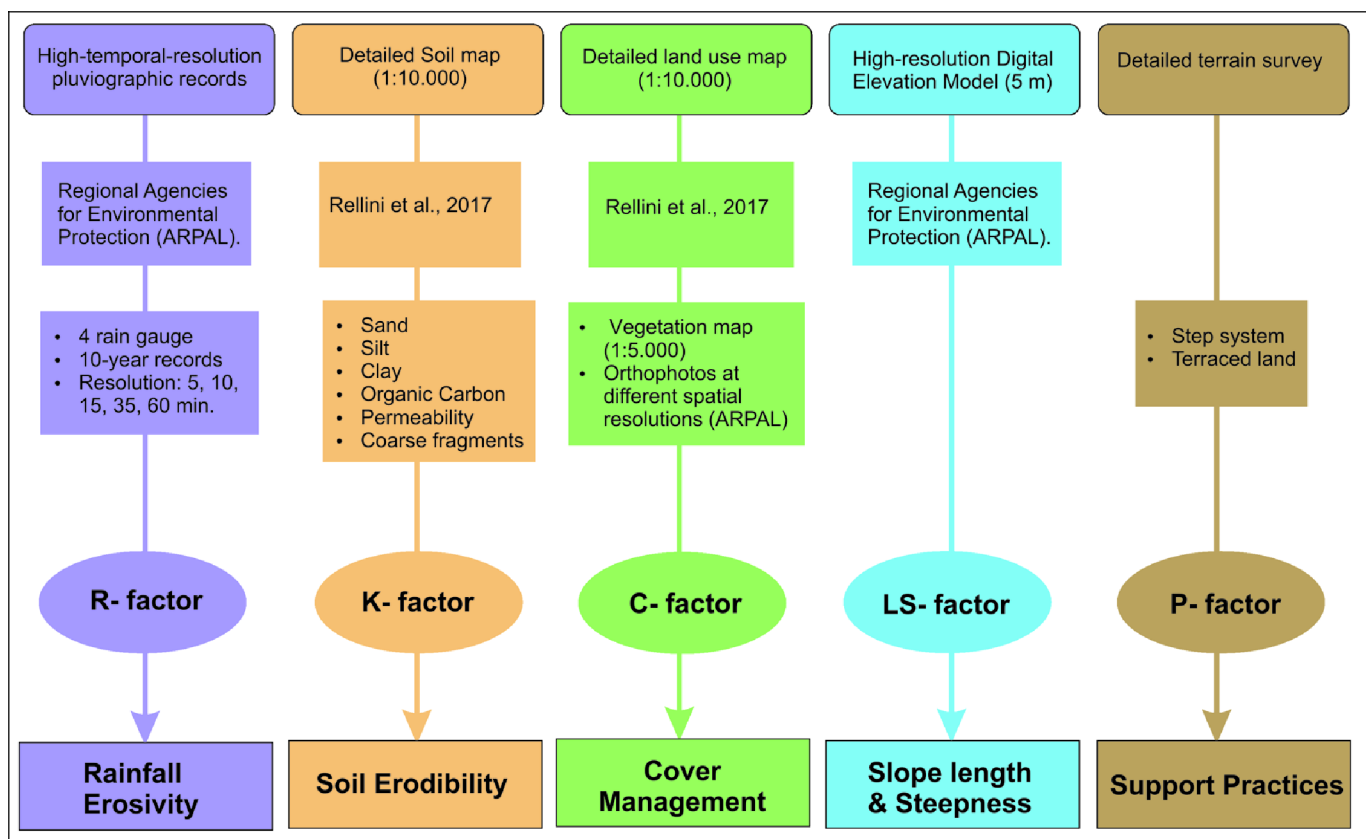
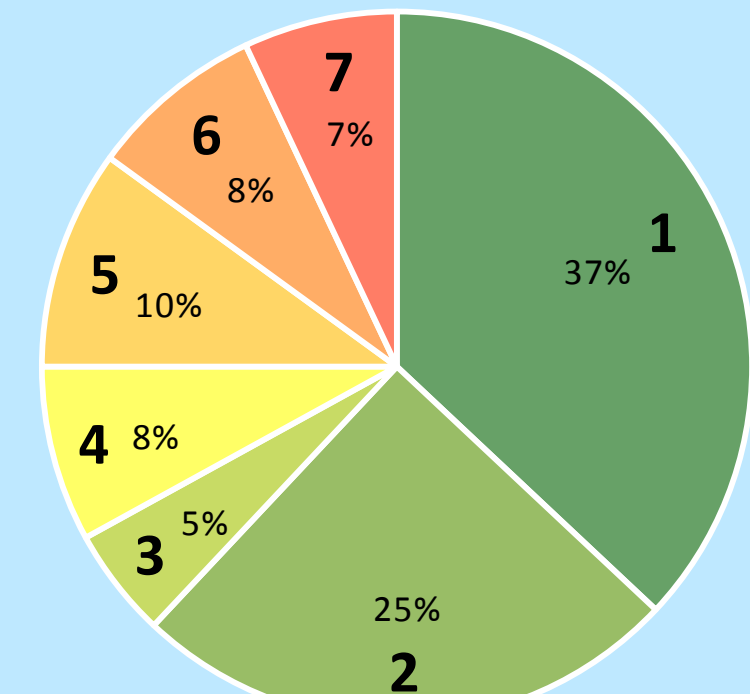
$$\text{Soil Loss} = R \text{ factor} * K \text{ factor} * LS \text{ factor} * C \text{ factor} * P \text{ factor}$$



SOIL LOSS (ton ha⁻¹ y⁻¹)

- 1 0 - 1.0 (Very low)
- 2 1.1 - 3.0 (Low)
- 3 3.1 - 5.0 (Moderate)
- 4 5.1 - 10.0 (High)
- 5 10.1 - 20.0 (Severe)
- 6 20.1 - 40.0 (Very severe)
- 7 > 40 (Extreme)

Erosion Classes



Input dataset used for estimation of soil loss factors for study area in RUSLE

Soil loss values range between 0 and 225 t ha⁻¹ yr⁻¹ with a mean value of 9 t ha⁻¹ yr⁻¹. According to Pimentel et al. (1976), a threshold value for a sustainable soil loss as is amounting to 11-12 t ha⁻¹ yr⁻¹. In the study area only about 25% of the cells show a soil loss value greater than the limit of "tolerable" soil loss. The highest soil loss values are associated with the cultivated areas on the terraced slopes. Hence, substantial conservation practices should be taken into account in these areas. The lack of maintenance of the existing practices (terracing and step type landscapes) could be a primary factor favoring soil erosion processes (photo 1). Moreover, possible cutting of the forest would cause serious consequences in terms of soil erosion in areas that show highest K-factor value on very steep slope (photo 2). The excessive deforestation due to the demand for raw materials and timber during historical times, for instance, lead to significant soil erosion increasing also on the summit area (photo 3). These observations are important for the present land cover preservation strategies.

